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To what extent has sustainable intensification in England been achieved?

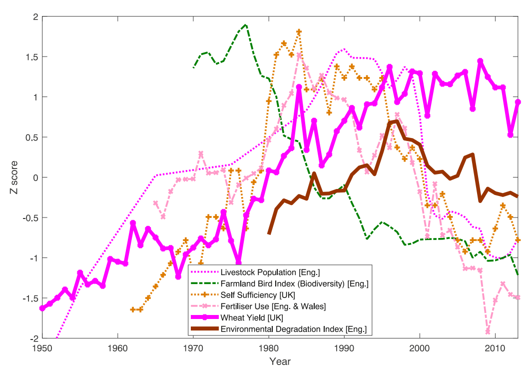
David I. Armstrong McKay [a,*], John A. Dearing [a], James G. Dyke [a], Guy M. Poppy [b], Les G. Firbank [c]

[a] Geography and Environment, University of Southampton, Southampton SO17 1BJ, UK
[b] Centre for Biological Sciences, University of Southampton, Southampton SO17 1BJ, UK
[c] School of Biology, University of Leeds, Leeds LS2 9JT, UK

HIGHLIGHTS

- Sustainable Intensification (SI) is needed to reduce farming's environmental impact.
- We assess the progress of SI in England by analysing agroecosystem service metrics.
- Degradation falls after ~1996 due to declining fertiliser use and livestock numbers.
- Low farm biodiversity and degradation 'offshoring' are major negative trade-offs.
- Future SI must address biodiversity loss, climate change, and offshored degradation.

GRAPHICAL ABSTRACT



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ABSTRACT

Agricultural intensification has significantly increased yields and fed growing populations across the planet, but has also led to considerable environmental degradation. In response an alternative process of 'Sustainable Intensification' (SI), whereby food production increases while environmental impacts are reduced, has been advocated as necessary, if not sufficient, for delivering food and environmental security. However, the extent to which SI has begun, the main drivers of SI, and the degree to which degradation is simply 'offshored' are uncertain. In this study we assess agroecosystem services in England and two contrasting sub-regions, majority-arable Eastern England and majority-pastoral South-Western England, since 1950 by analysing ecosystem service metrics and developing a simple system dynamics model. We find that rapid agricultural intensification drove significant environmental degradation in England in the early 1980s, but that most ecosystem services except farmland biodiversity began to recover after 2000, primarily due to reduced livestock and fertiliser usage decoupling from high yields. This partially follows the trajectory of an Environmental Kuznets Curve, with yields and GDP growth decoupling from environmental degradation above ~£17,000 per capita per annum. Together, these trends suggest that SI has begun in England. However, the lack of recovery in farmland biodiversity, and the reduction in UK food self-sufficiency resulting in some agricultural impacts being 'offshored', represent major negative trade-offs. Maintaining yields and restoring biodiversity while also addressing climate change, offshored degradation, and post-Brexit subsidy changes will require significant further SI in the future.

1. Introduction

Agriculture is already one of the leading drivers of environmental degradation around the world (Rockström et al., 2009, 2017; Steffen

* Corresponding author at: Stockholm Resilience Centre, Stockholm University, Kräftriket 2B, SE-10691 Stockholm, Sweden.
E-mail address: david.armstrongmckay@su.se (D.I. Armstrong McKay).

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et al., 2015; Tilman et al., 2001; Vitousek et al., 1997), yet global demand for food is forecast to continue to increase as the world's population grows to around 11 billion by the end of the 21st Century (UN Population Division, 2017). Sustainable Intensification (SI), whereby more food is produced per unit area but with a smaller environmental footprint, is a necessary (albeit not sufficient) means of tackling this challenge (Baulcombe et al., 2009; Firbank et al., 2013b; Garnett et al., 2013; Godfray and Garnett, 2014; Mahon et al., 2017; Poppy et al., 2014a, 2014b; Pretty, 1997; Thiaw et al., 2011; Tilman et al., 2011). SI implies a reduction in environmental degradation while food production continues to increase as a result of resource use decoupling from production. This process is likely to generate a type of Environmental Kuznets Curve (EKC) – with degradation peaking and then declining beyond a certain level of prosperity (Grossman and Krueger, 1995) – for those ecosystem services considered important for keeping regional socio-ecological systems within a safe operating space (Dearing et al., 2014). It has been claimed that at least some individual British farmers have achieved SI in recent years (Firbank et al., 2013b). Here we ask whether ecosystem services associated with UK agriculture at the regional scale are displaying SI or EKC behaviour, and what this means in terms of its sustainability.

Our approach is to identify trends of environmental degradation, ecosystem services, and socioeconomic factors linked to farming based on a wide range of regional agricultural and environmental data, prior to performing multivariate data analysis and developing a simple system dynamics model of the agricultural socio-ecological system. We use the Ecosystem Services framework, in which natural processes are conceptualised as providing services that benefit human wellbeing (Carpenter et al., 2009; Millennium Ecosystem Assessment, 2005). These in turn can be split into regulating (e.g. water quality, soil stability), provisioning (directly harvested, e.g. food, water, timber), and cultural (e.g. recreation, aesthetics) services. Also, under the Natural Capital framework, the metrics we quantify can be thought of as the condition of 'Assets' from which services are derived (Natural Capital Committee, 2017). We follow (Zhang et al., 2015), who used time-series of social, economic, and ecological conditions from the Lower Yangtze River Basin to develop aggregated indices of provisioning and regulating ecosystem services during the 20th Century. Regulating and cultural ecosystem services in this example included soil stability, biodiversity, air quality, sediment regulation, and sediment quality deduced from limnological records in the region (Dearing et al., 2012), while the yield of various different crops were used to represent provisioning ecosystem services, and records of parameters such as population growth and GDP used to indicate the socioeconomic aspects of the agroecosystem. For this part of China, there were clear negative trade-offs between increasing provisioning and declining regulating services with no strong evidence for decoupling between economic growth and environmental degradation as implied by the later stages of the EKC (Dearing et al., 2012, 2014; Zhang et al., 2015). Thus, the methodology of developing a wide range of ecosystem service metrics and performing multivariate data analysis offers an effective means of assessing the degree of sustainability of SI within an agroecosystem.

The UK experienced strong intensification in both arable and pastoral lowland agriculture after the 1960s during the second half of the 20th Century (Chamberlain et al., 2000; Firbank et al., 2008), while many ecosystem services became degraded, including farmland biodiversity, river water quality, and atmospheric emissions (Firbank et al., 2011). More recently, food production has tended to plateau, while some of the environmental degradation has been reduced (Firbank et al., 2011, 2013a, 2013b), even though overall UK economic growth has continued. Previous studies of SI in the UK have assessed ecosystem service trends and trade-offs on a national scale (Firbank et al., 2011, 2013a) and on a farm scale (Firbank et al., 2013b), but have not included testing for an EKC, multivariate data analysis, or model development.

In this study we have identified and assembled empirical time-series that summarise the post-1950 social, environmental, and economic performance of English agriculture in terms that can be related to the concepts of ecosystem services and the safe operating space for agroecosystems (Dearing et al., 2014). As well as analysing England as a whole, two sub-regions of England were selected to focus on differing farming systems: Eastern England for lowland arable agriculture and South-Western England for lowland pastoral agriculture (Morton et al., 2011). The objectives are: 1) to compare the trends in the English agroecosystem and two contrasting sub-regions since 1950 and identify their inter-relationships and possible drivers; 2) to test for the presence of an EKC between environmental degradation and economic growth compared with yields; and 3) to develop a simple system dynamics model of the English agroecosystem to identify potential means to influence the system towards a more resilient and sustainable state.

2. Material and methods

2.1. Data sources and processing

We searched for datasets from publically available sources that represented key agroecosystem services, including provisioning, regulating, and cultural services as well as socioeconomic performance. Annual data on the structure and economics of English agriculture were taken from the UK Department for Environment, Food, and Rural Affairs (DEFRA); environmental data were taken from sources such as the Environment Agency and limnological records; and socioeconomic data were taken from sources including the Office of National Statistics (Table 1). We sought the longest possible records available at an annual resolution, and used linear interpolation (Matlab, interp1 (The MathWorks Inc., 2016)) where necessary to cover data gaps. The acquired datasets were standardised as Z-score time-series in order to characterise relative changes rather than absolute changes over time (Fig. 1). Aggregated indices for River Nutrient Contamination (mean nitrate and phosphate concentrations), Environmental Degradation Index (EDI: the mean of river nutrient contamination, atmospheric non-greenhouse emissions, estimated soil erosion, and farmland bird index), Livestock Outputs (total meat and dairy products, excluding poultry), and an Estimated Soil Erosion Index (the difference between riverine suspended solids and biological oxygen demand) are calculated from average standardised values in order to give an overview of the behaviour of related variables. Phase plots were used to further explore the relationships between key variables and indices over time (Fig. 2 & Fig. 3). Detrended correspondence analysis (DCA; R, vegan, decorana (Oksanen et al., 2017; R Foundation for Statistical Computing, 2016)) and principal component analysis (PCA; R, prcomp) were also used to further investigate long-term trends in the data (Supplementary Figs. S8 & S9, Section S6 for R commands). Following this, 17 key parameters for the English agroecosystem were used for correlation analysis (Table 2 & Supplementary Fig. S10; R, PerformanceAnalytics, chart.Correlation (Peterson and Carl, 2014)) in order to identify, quantify, and categorise significant correlations. From this we use expert judgement and the literature to identify correlations that are hypothetically causal for use in the conceptual model (Fig. 4 & Fig. 5). Additional plots for climatic data, agricultural areas, and regional analyses repeated for Eastern and South-Western England are presented in the Supplementary material (Figs. S1–S7, S11–S18).

2.2. Data limitations

Regional analysis of the data is limited by both spatial and temporal resolution, and the mixture of regional and national-scale data available. The length of the aggregated EDI is limited by the unavailability of many datasets before ~1980. Data for farm subsidies, farm income, intermediate consumption, atmospheric emissions, and farmland biodiversity are currently only available either for England or the UK as a whole, and so for the regional analyses national-level data were used for these variables alongside regional-level data where available (see Table 1 and

Table 1
Study datasets, including description, data type, data coverage, time period, and data source.

Code	Metric	Description	Index type	Coverage	Time	Source
Yl	Yield (food provisioning)	Wheat yield (t/Ha)	Eco. service: provisioning	UK	1885–2014, (annual)	Cereal Production Survey[c] (DEFRA, 2014b)
Li	Livestock outputs (food provisioning)	Livestock outputs (meat & dairy); population	Eco. service: provisioning	England, counties	1973–2013 (annual); 1900–2013 (semi-decadal)	June Census of Agriculture[c] (DEFRA, 2014c)
Ae	Atmospheric emissions (non-GHG) (air quality)	Ammonia, PM, NMVOCs, & carbon monoxide (kt) [GHG separate]	Eco. service: regulating/cultural	England	1980–2013 (annual) [GHG: 1990–2013]	NAEI (Salisbury et al., 2015)
Rn	River nutrient contamination (water quality)	Mean nitrate and phosphate concentrations in river regions (mg/l)	Eco. service: regulating/cultural	Hydrological regions, monitoring stations[a]	1980–2013 (annual)	(Environment Agency, 2014)
Se	Extrapolated soil erosion (soil stability)	Difference between riverine suspended solids and BOD	Eco. service: regulating/cultural	River monitoring stations[a]	1980–2013 (annual)	Extrapolated from (Environment Agency, 2014)[c]
Bd	Farmland biodiversity	Farmland Bird Index (as proxy for wider biodiversity)	Eco. service: regulating/cultural	England	1970–2013 (annual)	RSPB, BTO (DEFRA, 2015a, 2016a)
Fu	Fertiliser usage	Total phosphate & nitrate usage by farms (kt)	Farm socio-economics	England & Wales	1965–2013 (annual)	British Survey of Fertiliser Practice[c] (DEFRA, 2014a)
Pu	Pesticide usage	Total usage on arable crops (weight applied by tonne)	Farm socio-economics	GB/UK	1974–2014 (irregular)	(FERA PUS Stats, 2015; Garthwaite, Unpublished)
Lb	Farm labour	Total labour headcount on farms (1000s)	Farm socio-economics	England, counties	1950–2013 (irregular)	June Census of Agriculture[c] (DEFRA, 2014c)
Fi	Farm income	Total UK farm income (real-term, aggregated, £)	Farm socio-economics	UK	1973–2013 (annual)	Total Income from Farming[c] (DEFRA, 2015c)
Fs	Farm subsidies	Total UK/EU subsidies to all UK farms (real-term, £)	Farm socio-economics	UK	1973–2013 (annual)	Total Income from Farming[c] (DEFRA, 2015c)
Ic	Input costs (intermediate consumption)	Total input spending by all UK farms (real-term, £)	Farm socio-economics	UK	1973–2013 (annual)	Total Income from Farming[c] (DEFRA, 2015c)
Po	Population	Total population by area (1000s)	UK socio-economics	England, regions	1851–2014 (decadal <1981, annual since)	(ONS, 2015a)[d]; (Great Britain Historical GIS Project, 2015)[b]
Ct	Climate – temperature	Yearly average temperature (°C)	Environment context	England, regions	1910–2015 (annual)	(Met Office, 2015)
Cr	Climate – rainfall	Yearly total rainfall (mm) [& riverflow reconstruction]	Environment context	England, regions	1910–2015 [~1865–2002] (annual)	(Met Office, 2015); CRU, UEA (Jones et al., 2004)
Fp	Food prices	Global food price index (real-term)	UK socio-economics	Global	1961–2015 (annual)	(UN FAO, 2015)
Gdp	GDP per capita	Real GDP/cap. (£/cap., CVM market prices, SA)	UK socio-economics	UK	1955–2014 (annual)	UK Economic Accounts[d] (ONS, 2015b)

[a] Hydrological Region stations: Anglian: Bedford Ouse; SW: Exe (plus Tamar for England average); SE: Medway & Thames; Midlands: Severn & Trent; NE: Aire, Don, Tees, & Tyne; NW: Dee, Mersey, & Ribble.
[b] This data is provided through www.VisionofBritain.org.uk and uses statistical material which is copyright of the Great Britain Historical GIS Project, Humphrey Southall and the University of Portsmouth.
[c] Crown copyright 2017. Adapted from data from the Department for Environment, Food and Rural Affairs under the Open Government Licence v.3.0 (http://www.nationalarchives.gov.uk/doc/open-government-licence/version/3/).
[d] Crown copyright 2017. Adapted from data from the Office for National Statistics licensed under the Open Government Licence v.3.0 (http://www.nationalarchives.gov.uk/doc/open-government-licence/version/3/).

Supplementary material for details of the spatiotemporal data coverage for each variable). We found insufficient data to quantify other key ecosystem services such as climate regulation, freshwater extraction, pest regulation, disease regulation, and pollination over the whole 1980–2013 period, and so these were not included in our analyses (Millennium Ecosystem Assessment, 2005).

Sediment regulation and soil erosion were difficult to constrain from the available hydrological and limnological records and no long-term high-resolution regional/national records of soil erosion are available, with most soil erosion studies providing spatial rather than temporal comparisons (e.g. Boardman, 2013). It was therefore necessary to extrapolate the suspended sediment in key rivers from the difference in Z-scores between suspended solids and algae population (the latter by using biological oxygen demand as a proxy). Although this provided a usable soil erosion metric, a direct metric of suspended sediment and/or sediment accumulation from lakes and rivers in large catchments in both regions would provide a more accurate and regionally representative record of sediment regulation. As a result we interpret extrapolated soil erosion trends cautiously. The agricultural atmospheric emissions data is based on modelling from known emission sources, and so is inherently linked to livestock and fertiliser data. This will upwardly bias the correlation between these variables, but there is high confidence in the veracity of this relationship (Salisbury et al., 2015).

We use the England Farmland Bird Index (FBI) (DEFRA, 2016a) as a proxy indicator for wider farmland biodiversity and abundance as it is the longest-running and highest-spatial resolution farmland-specific ecosystem index available. It closely resembles the overall trend of the UK Priority Species Abundance where the datasets overlap, and as many specialist farmland birds have an insectivorous diet their abundance is likely to be closely linked to insect availability and diversity (Benton et al., 2002; Fuller, 2000; Maron and Lill, 2005; Razeng and Watson, 2010, 2015). Other recent reports (Hayhow et al., 2016; Mathews et al., 2018) emphasise the wider declines in the abundances of farmland plants, vertebrates and invertebrates since the 1970s and 1990s. This means that the FBI is therefore only an indirect proxy for wider agroecosystem biodiversity, and a more comprehensive index or direct measurements may reveal differing trends (Lindenmayer and Likens, 2011). Woodland birds could also be included in the biodiversity index as part of the wider agriculture-dominated landscape, but here we exclude them in order to focus on only the species most directly impacted by agricultural processes.

We regard EDI as reflecting both regulating and cultural ecosystem services, with farmland biodiversity influencing wider ecosystem resilience as well as being of high societal value and pollution viewed negatively by society as well as affecting ecosystem regulation (Loos et al., 2014; Mace et al., 2012; MacFadyen et al., 2009; Srivastava and

Vellend, 2005). However, EDI does not reflect all regulating services, with insufficient data for the whole 1980–2013 period to include factors such as carbon emissions, soil organic carbon, water use, and pest regulation, while the biodiversity and soil erosion indices used in the EDI are also limited. Each source index for the EDI is also weighted equally, which may not reflect the differing importance of each for agroecosystem resilience but in the absence of further information equal weighting avoids prejudicing the index without an empirical basis. Strong trends in one sub-index may also mask important trends in another sub-index and give a misleading overall picture. Further work is needed to characterise the relative importance of the metrics of each ecosystem service to overall environmental degradation, and to fill in the data gaps where no long-term ecosystem service metric is currently possible.

3. Data analysis

3.1. English agroecosystem trends

Our results clearly illustrate the process of agricultural intensification in the English agroecosystem during the 1980s and 1990s coupled to contemporaneous degradation in ecosystem services, with a subsequent partial environmental recovery after the late 1990s that suggests the commencement of SI (Fig. 1 & Table 2; Supplementary Fig. S10). Rising wheat yields (and acreage, Supplementary Fig. S5) are linked to increasing fertiliser usage up until ~1984, which is driven by the introduction of new cultivars in the 1970s that could utilise higher nitrogen applications (Hawkesford, 2014), along with mechanisation and increased pesticide use (Firbank et al., 2011). Fertiliser use also increased on lowland grasslands (DEFRA, 2014a). However, high fertiliser usage is strongly correlated with high riverine nutrient contamination and atmospheric emissions due to the runoff, denitrification, volatilisation, and leaching of fertilisers after application. Increasing livestock output and population is also correlated to river nutrient contamination and atmospheric emissions through effluent runoff and enteric emissions. Together with sharp declines in farmland birds, which in our data is negatively correlated with yields and temperature, the aggregated EDI increased through to the mid-1990s.

The subsequent recovery in EDI is driven by the decoupling of fertiliser usage and yield, with wheat yields stable after 1984 despite a significant decline in fertiliser usage (in particular of phosphate in arable areas) (Fig. 3). This reflects improved farming practice in the targeted application of fertiliser in response to new regulations such as the introduction of Nitrate Vulnerable Zones in 1998–2002, knowledge exchange with academic and advisory bodies (such as the Agriculture and Horticulture Development Board), and increasing fertiliser prices (Firbank et al., 2011), as well as a reduction in cattle numbers and increased manuring efficiency specifically reducing nitrate application on grassland (DEFRA, 2014a). Consequently, there is a reduction in the contamination of rivers by fertiliser runoff and a decline in atmospheric emissions, aided by the rapid drop in livestock numbers in the 2000s (partially due to the 2001 foot-and-mouth disease outbreak and subsidy reform) and the banning of field burning in 1993. Stagnating yields have been linked to the growing impact of climate extremes and changes in rotation practices (Brisson et al., 2010; Knight et al., 2012).

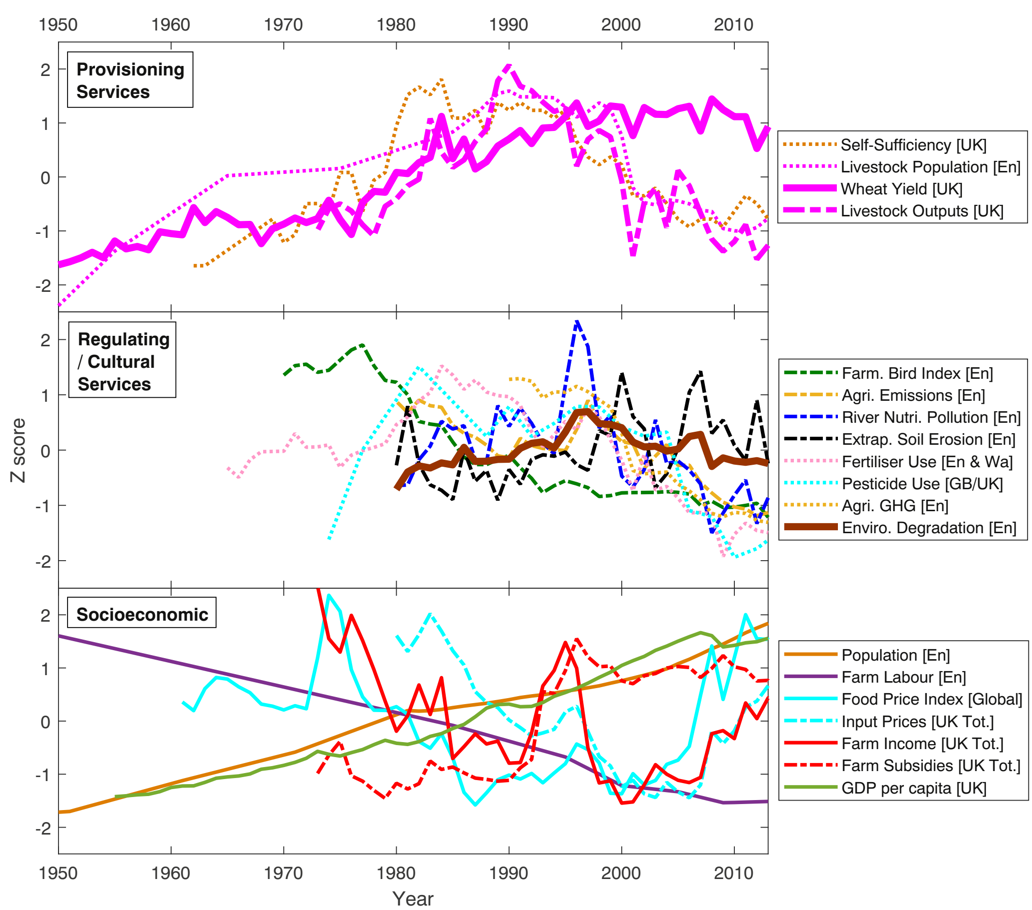

Fig. 1. Z-score plot illustrating the evolution of the English agroecosystem as reflected by indices of: a) provisioning ecosystem services, b) regulating/cultural ecosystem services, and c) socioeconomic parameters. Climate data and regional variations are illustrated in Supplementary Figs. S1–3, S7, S11, & S15. See text for detail.

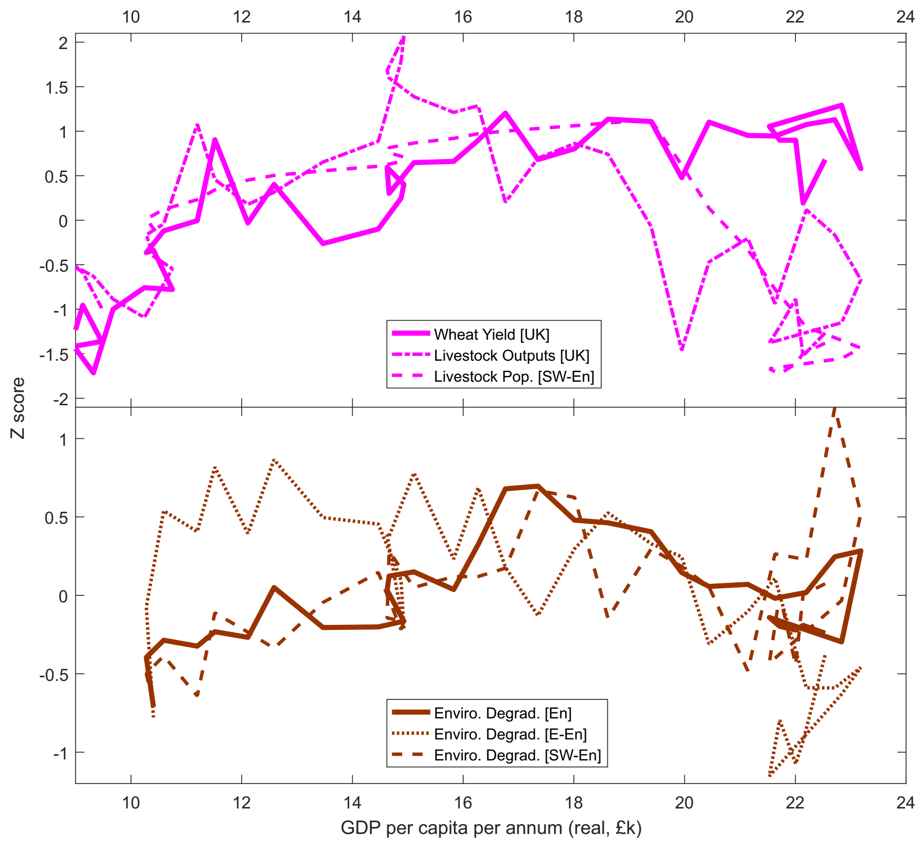

Fig. 2. Phase plots of provisioning services (top) and the environmental degradation index (bottom) in England and the sub-regions of Eastern and South-Western England versus UK GDP per capita per annum.

In contrast to the improvements in river and atmospheric pollution, farmland biodiversity failed to recover after the initial rapid decline in the early 1980s despite improvements in river nutrient contamination and atmospheric emissions. This suggests that the drivers of farmland biodiversity decline are different from the drivers of river nutrient contamination and atmospheric emissions, and have been hypothesised to be linked to factors such as sowing timing, grassland improvement, habitat diversity, and livestock stocking density (Benton et al., 2003; Butler et al., 2007; Chamberlain et al., 2000; Firbank et al., 2008; Fuller, 2000; Krebs et al., 1999; Newton, 2004). A gradual increase in 'land sparing' in England since 1950 (Supplementary Fig. S5), potentially linked to intensification on productive land making marginal land less economically viable and therefore more suitable for 'sparing' for conservation purposes (Balmford et al., 2005, 2015; Ewers et al., 2009; Green et al., 2005; Phalan et al., 2016), has not compensated for overall farmland biodiversity decline. This may be due to sparing mostly taking place from rough grazing land in upland regions and low-yielding common land rather than from more intensive arable or pastoral lowland areas, and so has not directly benefited the wildlife specifically dependent on the latter for which the FBI acts as a proxy for. However, the expansion of agri-environment schemes such as set-aside land in the early 1990s and environmental stewardship after set-aside was discontinued in 2005 does coincide with a reduced rate of decline in the FBI (DEFRA, 2015a). Farmland biodiversity is a key ecosystem service in the wider agroecosystem, and its continued decline undermines the overall SI trend (Baulcombe et al., 2009; Mace et al., 2012; Thiaw et al., 2011). This implies that despite some improvements the English agroecosystem has not yet reached a safe operating space, and that novel approaches to halting and reversing farmland biodiversity loss are required that are not included in the current SI process. In contrast to the other indices, the extrapolated Soil Erosion Index shows no discernible trend and only correlates with average yearly rainfall in the regional indices.

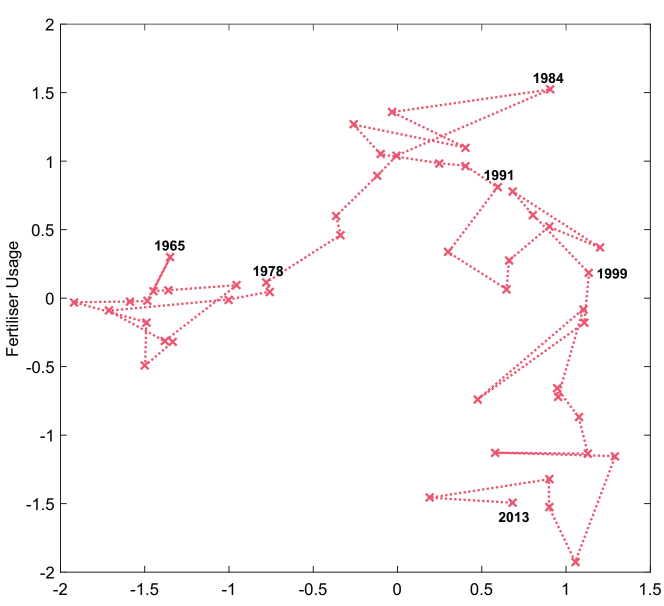

Fig. 3. Phase plot of the Z-scores for wheat yield (UK) and fertiliser usage (total for England and Wales) between 1965 and 2013.

Socioeconomic trends for the English agroecosystem tend to not correlate with as many variables as the biophysical variables in the

Table 2
Correlated variables from our correlation analysis hypothesised to represent causal relationships (or only possibly linked in italics) in the English agroecosystem and used to build the conceptual models. Correlation significance (p-value) is given as *** for p < 0.001, ** for p < 0.01, * for p < 0.05, — for p < 0.1, and N/A for p > 0.1.

Variable 1	Variable 2	Correlation		Hypothesised driver
Farmland biodiversity	Wheat yield	−0.72***	Strong negative	Landscape/ecosystem homogenisation
Livestock population	Atmospheric emissions	+0.51**	Strong positive	Livestock enteric emissions
Livestock population	River nutrient contamination	+0.66***	Strong positive	Livestock effluent runoff into rivers
Fertiliser usage	Atmospheric emissions	+0.80***	Very strong positive	Fertiliser degassing
Fertiliser usage	River nutrient contamination	+0.58***	Moderate positive	Fertiliser runoff into rivers
Fertiliser usage	Wheat yield	−0.57***	Moderate negative (false)	Initially positive, but decouples with SI to give net negative
Climate – temperature	Farmland biodiversity	−0.46**	Moderate negative	Heat stress on wildlife and forced migration
Climate – temperature	Wheat yield	+0.51**	Moderate positive	Lengthened growing season
Climate – rainfall	River nutrient contamination	−0.54**	Moderate negative	Pollution dilution in rivers
Climate – rainfall	Soil erosion	+0.67***	Strong positive	Sediment runoff during rainstorms
Farm subsidy	Wheat yield	+0.81***	Very strong positive (false/indirect)	Production incentivised by subsidies, but not directly causal
Intermediate consumption	Fertiliser usage	+0.60***	Strong positive	Declining fertiliser use saves farmers' money
Intermediate consumption	Farm income	+0.61***	Strong positive	Higher income allows higher input spending
Pesticide use	Wheat yield	−0.39*	Weak negative (false)	Initially positive, but decouples with SI to become net negative
Farm income	*Food prices*	*+0.29 N/A*	*Weak positive [insignificant]*	*High food prices tend to elevate incomes*
Farm subsidy	*Farm income*	*−0.12 N/A*	*Very weak negative [insignificant] (false?)*	*Subsidies support incomes*
Pesticide usage	*Farmland biodiversity*	*+0.69****	*Moderate positive (false)*	*Pesticides known to harm some species*
Farm subsidy	*Sustainable intensification*	*N/A*	*N/A*	*Hypothetical positive impact of subsidies on SI*

correlation analysis (Table 2). Wheat yield is strongly correlated with farm subsidies, which reflects increased direct subsidies to farmers after 1992 coinciding with elevated yields, and does not imply causation. High farm income and fertiliser usage correlate with higher intermediate consumption (i.e. total farm spending), and high food prices correlate with lower livestock outputs. Total farm income appears to partially follow trends in both Food Price Index and farm subsidies (Fig. 1) but is not significantly correlated with either. Despite a general increase in total farm income from a minimum in 2000, by the end of our study period ~46% of UK farms failed to recover their costs in that year and therefore remain heavily dependent on subsidies (DEFRA, 2015b). This reliance on EU subsidies results in income fluctuations following the sterling-euro exchange rate (e.g. the drop in subsidies in 2014 (DEFRA, 2015b)) and could lead to major changes in income during and following the UK's withdrawal from the EU ('Brexit'). As a result, future SI needs to incorporate the changing role of subsidies and ensure the financial security of farmers. No directly causative correlations were found with farm labour headcount, with continuously declining employment strongly anti-correlating with GDP per capita growth. This decline reflects continued agricultural modernisation and mechanisation, with growing national wealth associated with a peak and then a decline in the proportion of UK GDP and labour force involved in agriculture.

These trends are also supported by both the DCA (Supplementary Fig. S8) and PCA (Supplementary Fig. S9) results. Most of the data variance lies in the first axis (DCA1: eigenvalue of 0.2278, axis length of 1.5235; PC1: ~54% of variance) and shows a shift from an initial state associated with lower yields, high inputs (including labour and fertiliser), high river/atmosphere pollution (linked to inputs, e.g. high fertiliser use, and livestock), higher income, lower subsidies and food prices, to a new state with higher yields, lower inputs and river/atmosphere pollution, lower farm employment and income, and higher subsidies and food prices. We interpret this as primarily reflecting both the modernisation and commencement of sustainable intensification of English agriculture during the study time-period. There are also contributions to DCA1 and PC1 from increasing population, increasing temperatures, and the continual deterioration of farmland biodiversity. DCA2 and PC2 explains much less of the data variance (DCA2: eigenvalue of 0.03905, axis length of 0.81272; PC2: ~16% of variance) and have differing contributions from each variable with no obvious overall interpretation. DCA2 is notable though for the strong opposition of river nutrient contamination versus rainfall and soil erosion, which could potentially arise from high rainfall years being associated with diluted contamination but higher soil erosion.

3.2. Regional differences

Regional Z-score time-series, PCA, DCA, and correlation analyses show that the trends and correlations of the key variables of the Eastern England and South-Western England agroecosystems are mostly similar to the all-England analyses, but that there are some differences. In contrast to the all-England analysis, in our extrapolated soil erosion index arable Eastern England experiences relatively high soil erosion rates during the 1980s and early 1990s followed by a decline, while pastoral South-Western England appears to have had overall increasing soil erosion rates since the early 1990s (Supplementary Figs. S11 & S15). Eastern England also experiences an earlier and higher peak in environmental degradation before subsequently showing a stronger recovery than the rest of England, and rainfall trends also do not correlate as well with other variables in Eastern England (Supplementary Figs. S14 & S18). Regional PCA and DCA results are mostly similar to the all-England PCA results, with PC1 containing similar trends and accounting for ~62% and ~54% of the data variance in Eastern and South-Western England respectively, and PC2 explaining an additional ~14% and ~16% respectively (Supplementary Figs. S12–S13 & S16–S17). However, in Eastern England soil erosion increases with positive PC1 values reflecting the gradual reduction in soil erosion over time in contrast to all-England, and PC2 also reflects higher rainfall and temperature along with higher atmospheric emissions, wheat yield, and soil erosion in the negative direction. Eastern England differs more from all-England than South-Western England in all analyses, which along with the rainfall and soil erosion trends we suggest is because most of England more closely resembles the mixed and pastoral farming of South-Western England with higher rainfall and more variable topography (falling in the larger Celtic broadleaf forest WWF ecoregion) than the intensive arable agriculture concentrated in drier and lower-lying Eastern England (mostly falling in the smaller English lowland beech forest WWF ecoregion) (Morton et al., 2011; Olson et al., 2001). Eastern England's earlier and higher peak in environmental degradation implies the rapid intensification of arable agriculture in this region had stronger impacts than in South-Western England, but that these impacts have now mostly abated. However, our analysis does not include more novel

impacts of intensive agriculture, such as recent evidence of potentially harmful levels of riverine neonicotinoid (a controversial insecticide) contamination clustered in Eastern England (Shardlow, 2017).

4. Environmental Kuznets Curves and degradation 'offshoring'

Environmental degradation appears to follow the trajectory of an EKC in both the whole English agroecosystem as well as in both Eastern and South-Western England. Both wheat yield and degradation increase up to UK GDP per capita per annum of ~£17,000 before degradation declines with further increases in GDP while wheat yields stabilise (although livestock declines as a result of the 2001 foot-and-mouth outbreak, see Section 3.2) (Fig. 2). As a result, environmental degradation in the English agroecosystem partially follows a classic EKC trajectory (Dinda, 2004), with soil, air, and water degradation (but not biodiversity) rising with economic development before declining past a critical threshold as more efficient technologies and practice (e.g. one-pass systems, new crop varieties, integrated pest management (Baulcombe et al., 2009)) and environmental regulation (e.g. Nitrate Vulnerable Zones) are established. The gap between falling environmental degradation and stable yields relative to GDP (Fig. 2) provides clear evidence that some degree of SI has taken place, as yields have been maintained with a smaller environmental footprint while overall prosperity has continued to grow. However, this overall trend is not reflected by farmland biodiversity, which continues to decline despite economic growth and so displays no Kuznets Curve behaviour itself. SI tends to be associated with greater resource use efficiency, which can generate a cleaner environment but not necessarily a more biodiverse one (Firbank, 2005). Additionally, having increased to a peak in the early 1980s with intensification since the mid-1990s UK agricultural self-sufficiency has declined from ~74% to ~60% for all food (or ~85% to below 75% for just indigenous-type food) (DEFRA, 2016b), indicating that some of the UK's agricultural impact has effectively been offshored to other agroecosystems as a result of globalisation (Fig. 1 & Supplementary Fig. S6). This implies that environmental degradation may not have declined so much or at all if the UK had maintained or increased self-sufficiency in food production between 1980 and 2013. Together with poor biodiversity trends, this indicates that only partial SI has been achieved in the UK in this time, and that in order to reach both regional and global safe operating spaces for agroecosystems future SI will need to both halt biodiversity loss and ensure damaging practices are not simply offshored to poorer countries with weaker regulations. On the regional scale, degradation in South-Western England matches the trajectory of all-England fairly closely (despite an apparent resurgence in EDI in 2006 due to anomalously and potentially unreliably high extrapolated soil erosion in the River Exe), whereas environmental degradation in Eastern England occurs more rapidly and then subsequently improves by a greater degree than South-Western or all-England. This further illustrates the greater and more rapid impact on the environment of arable intensification versus the intensification of mixed or pastoral farming elsewhere.

5. Conceptual modelling

5.1. Model development

Following the data analysis we developed a simple system dynamics model using the Vensim PLE platform (Ventana Systems Inc., 2015) in order to further evaluate our understanding of the relationships within the English agroecosystem and the impacts of the changing nature of intensification between 1980 and 2013. Simple system dynamics models are a useful way to rapidly explore our understanding of a dynamical system using relative trends rather than absolute quantities (e.g. Meadows, 2008; Meadows et al., 1972). We restricted the relationships in the model to those that are both: a) commonly proposed as causative in the literature and from expert judgement, and b) showed statistically significant correlations in our dataset (Table 2 & Supplementary Fig. S10), in order to exclude spurious correlations. We use simple linear relationships and approximated trends of fertiliser usage, livestock population, temperature, rainfall, farm subsidy, and farm income in order to drive changes in farm biodiversity, yield, atmospheric emissions, soil erosion, river nutrient contamination, and input spending for the 1980–2013 period (Fig. 4). Each variable changes according to the averaged changes of its input variables – for example, changes in River

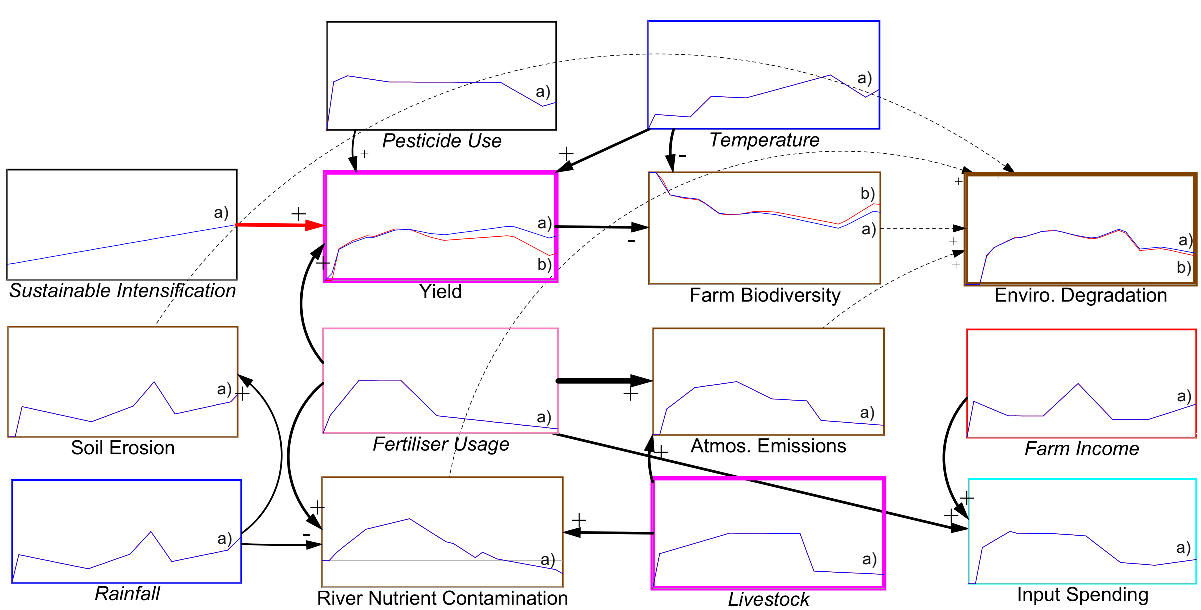

Fig. 4. Simple system dynamics model for the English agroecosystem, with simulation drivers/results for 1980–2013 shown in each variable box (italics for imposed drivers). Scenarios include: a) with (blue lines) and b) without (red) SI. Arrow thickness indicates correlation strength, dotted arrows show drivers of the Environmental Degradation Index, the dashed arrow shows the hypothesised effect of Sustainable Intensification), arrow symbols indicate correlation type (positive [+], negative [−], or variable [x]), and box colours match the colours used in the Z-score plots (Fig. 1; thick-lined boxes match thicker Z-score lines). Created using Vensim PLE (Ventana Systems Inc., 2015).

Nutrient Contamination are the average of the changes in Fertiliser Usage, Livestock, and Rainfall – and assumes equal weighting for each input. This assumption is likely to be inaccurate as some factors will be more important than others, but in the absence of further information we assign equal weightings as a starting point. There are several factors missing from this model which we exclude due to a lack of full datasets or direct correlations, such as level of mechanisation and food prices. There are no closed loops in this model, and so no feedback loops are expected to operate.

The model successfully recreates the trends in the non-driver variables for this time period (Fig. 1 & Fig. 4), with yield increasing and then plateauing, farm biodiversity declining and then plateauing, both river nutrient contamination and atmospheric emissions peaking and then declining as fertiliser use and livestock populations peak, soil erosion staying fairly level in the long-term, and input spending dropping in the 1990s. Yield is dependent on a normative 'Sustainable Intensification' variable that we introduce, which has to constantly increase in order to offset the impact of declining fertiliser usage. In this context SI represents improved fertiliser application practices and other improvements in crop management, but is not represented by a direct data proxy in our analysis and so has an imposed linear increase over time. Removing this SI variable results in yield peaking and then declining in line with fertiliser usage.

5.2. Future projections

In order to use the system dynamics model for future scenario exploration further hypothetical relationships that are likely to play a role in affecting future trends are added to the model (shown by the red arrows and variables in Fig. 5 and based on the possibly linked causal relationships in Table 2) and projected trends for model drivers imposed, including consistently increasing temperature, an erratic rainfall trend, stable subsidies (uncertain in a post-Brexit context), and stable but high food prices (Fig. 5). While mean annual temperature and yield are positively correlated in our data between 1980 and 2013, it is likely that further temperature increases will begin to reverse this correlation in the future and so we model further temperature increases to have net negative impacts on yields. We have also introduced estimated variables such as mechanisation for which full datasets were not available, for which we have estimated their past long-term trends. Based on this we explore several future scenarios featuring different responses to exogenous forcing such as increasing temperature and increasing variance in rainfall (Fig. 5).

In the 'Continual SI' scenario we allow SI to improve at a constant rate (increasing by a further 112% more than the 1980–2013 improvement), which counteracts the negative impact of increasing temperature, stabilises biodiversity loss, and reduces soil erosion despite consistent levels of mechanisation. If SI is instead kept fixed at 2013 levels until 2050 ('No Further SI' scenario), yields begin to fall and improvements are not observed after 2013 in the latter variables. In order to allow biodiversity to gradually recover while yield remains stable ('Biodiverse SI' scenario) it is necessary to reduce mechanisation and pesticide use to ~73% and ~20% below 2013 levels respectively while significantly increasing SI (to 200% more than the 1980–2013 improvement). Increasing yield beyond current levels rather than allowing it to plateau indefinitely ('Maximise Yield' scenario) requires some combination of this accelerated SI and increased mechanisation (by 75%), fertiliser usage (to previous peak), or pesticide use (to previous peak), but increasing these latter variables also reverses the recovery in fertiliser pollution and forces biodiversity into dangerous decline. Allowing a gradual recovery in livestock population to previous peak levels in conjunction with 'Continual SI' ('Livestock Intensification' scenario) results in partial reversals to the recoveries in atmospheric emissions and river nutrient contamination, although neither reaches the levels seen in the 1980s unless fertiliser use also increases.

These results suggest that it is difficult to both increase yield or livestock population and limit environmental degradation and further biodiversity decline without continual and significant improvements in SI. However, the SI variable is a significant simplification of a complex set of decisions, processes, and impacts surrounding farming practice with no upper limits, and it cannot be assumed that SI can consistently increase in order to offset other negative pressures on yield and biodiversity. Further work to better understand these dynamics is needed.

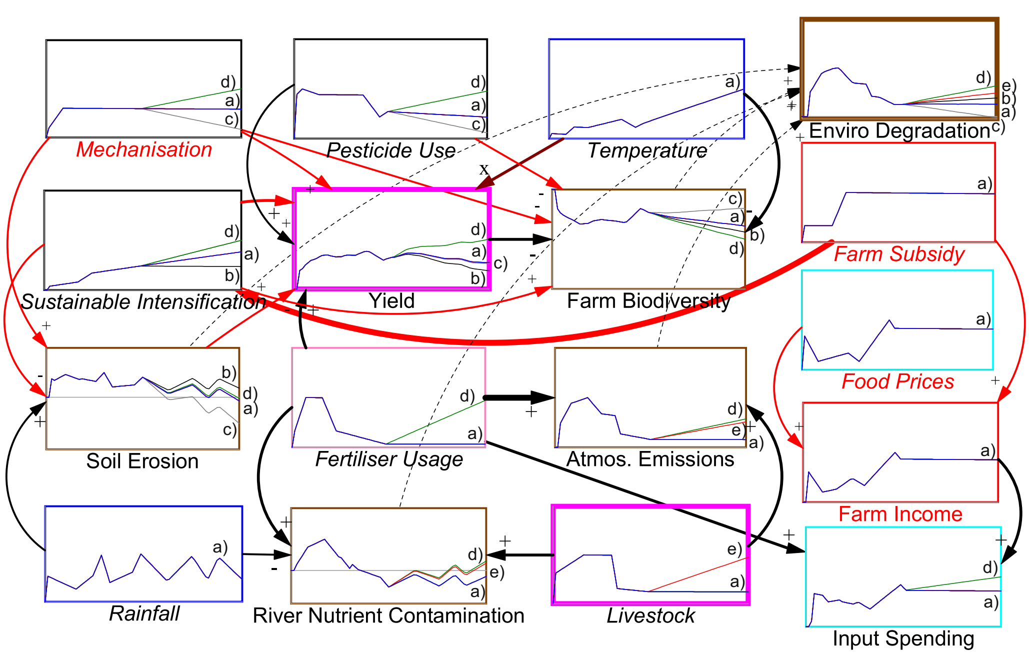

Fig. 5. Extended simple system dynamics model for the English agroecosystem, with simulation drivers/results for 1980–2050 in different scenarios. Scenarios include: a) 'Continual SI' (blue lines), b) 'No Further SI' (black), c) 'Biodiverse SI' (grey), d) 'Maximise Yield' (green), and e) 'Livestock Intensification' (red). Arrow and box weights and symbols are as in Fig. 4, except red text which indicates variables not included in the 1980–2013 model. Created using Vensim PLE (Ventana Systems Inc., 2015).

6. Conclusions

In this study we use publicly available data to construct metrics assessing the impact of agricultural intensification on environmental degradation in the English agroecosystem and use a simple system dynamics model to analyse future scenarios. From these analyses it is clear that agricultural intensification drove increased environmental degradation in England during the 1980s. In the 1990s fertiliser and pesticide usage decoupled from high yields with a reversal in the degradation of several ecosystem services (e.g. river nutrient contamination and atmospheric emissions), suggesting that SI began to take place. When plotted against GDP per capita this process follows an Environmental Kuznets Curve, suggesting better environmental protection with greater prosperity. Despite an increase in land sparing, farmland biodiversity has not experienced any recovery making it the major negative trade-off in current SI practices. Additionally, reduced agricultural self-sufficiency indicates some agricultural impacts may have been 'offshored' abroad. These two outcomes undermine attempts to achieve future English and global SI, and indicate that English agroecosystems have not yet reached a safe or just operating space. Similar patterns are observed in both arable-dominated Eastern England and pastoral-dominated South-Western England, although the impact of intensification was stronger in arable Eastern England. A simple system dynamics model of the English agroecosystem recreates the basic trends of several ecosystem services between 1980 and 2013 when assuming an increase in SI. The impacts of uncertain levels of subsidies post-Brexit and increasing climatic impacts were explored in future scenarios. These show that: maintaining or increasing yields and livestock populations while also restoring biodiversity; maintaining the environmental gains achieved since the 1990s; and improving the financial viability of farming, will all prove challenging. Further SI featuring novel policies and approaches to tackle current trade-offs – including reforms to subsidies and agri-environment schemes focusing on restoring biodiversity and reducing degradation offshoring – is required to meet these challenges, but the extent to which further intensification can also continue to become more sustainable remains uncertain.

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Author contributions

All authors designed the research; DIAM performed the data analysis and modelling; all authors contributed to the interpretation of the data analysis and modelling results; DIAM wrote the paper with input from all other authors; all authors gave final approval for publication.

Competing interests

The authors declare no competing interests.

Data availability

All data used in this study is available from publically accessible data sources cited in the text (see Table 1 for sources), with the minor exception of an as-of-yet unpublished extension to the pesticide usage dataset (provided on request by David Garthwaite and FERA PUS Stats) which provides critical extra context to the peak and decline of pesticide use in the UK.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2018.08.207.

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